5. Agriculture

Agricultural activities contribute directly to emissions of greenhouse gases through a variety of processes. This chapter includes the following sources: enteric fermentation in domestic livestock, livestock manure management, rice cultivation, agricultural soil management, and agricultural residue burning (see Figure 5-1). Agriculture-related land-use activities, such as conversion of grassland to cultivated land, are discussed in the Land-Use Change and Forestry chapter.

Figure 5-1: 1998 Agriculture Chapter GHG Sources

In 1998, agricultural activities were responsible for emissions of 148.4 MMTCE, or 8 percent of total U.S. greenhouse gas emissions. Methane (CH_4) and nitrous oxide (N_2O) were the primary greenhouse gases emitted by agricultural activities. Methane emissions from enteric fermentation and manure management represent about 19 and 13 percent of total CH_4 emissions from anthropogenic activities, respectively. Of all domestic animal types, beef and dairy cattle were by far the largest emitters of methane. Rice cultivation and agricultural crop residue burning were minor sources of methane. Agricultural soil management activities such as fertilizer application and other cropping practices were the largest source of U.S. N_2O emissions, accounting for 71 percent. Manure management and agricultural residue burning were also smaller sources of N_2O emissions.

Table 5-1 and Table 5-2 present emission estimates for the Agriculture chapter. Between 1990 and 1998, CH_4 emissions from agricultural activities increased by 19 percent while N_2O emissions increased by 12 percent. In addition to CH_4 and N_2O, agricultural residue burning was also a minor source of the criteria pollutants carbon monoxide (CO) and nitrogen oxides (NO_x).

Table 5-1: Emissions from Agriculture (MMTCE)

Gas/Source	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
CH_4	**50.2**	**50.8**	**52.1**	**53.3**	**56.2**	**57.4**	**57.6**	**59.1**	**59.5**
Enteric Fermentation	32.7	32.8	33.2	33.7	34.5	34.9	34.5	34.2	33.7
Manure Management	15.0	15.5	16.0	17.1	18.8	19.7	20.4	22.1	22.9
Rice Cultivation	2.4	2.3	2.6	2.4	2.7	2.6	2.4	2.6	2.7
Agricultural Residue Burning	0.2	0.2	0.2	0.1	0.2	0.2	0.2	0.2	0.2
N_2O	**78.8**	**80.0**	**81.8**	**81.0**	**87.4**	**84.3**	**86.4**	**88.3**	**88.0**
Agricultural Soil Management	75.3	76.3	78.2	77.3	83.5	80.4	82.4	84.2	83.9
Manure Management	3.4	3.6	3.5	3.7	3.8	3.7	3.8	3.9	4.0
Agricultural Residue Burning	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Total	**129.9**	**131.6**	**134.7**	**135.1**	**144.4**	**142.5**	**144.8**	**148.2**	**148.4**

Note: Totals may not sum due to independent rounding.

Table 5-2: Emissions from Agriculture (Gg)

Gas/Source	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
CH_4	**8,769**	**8,872**	**9,091**	**9,306**	**9,809**	**10,015**	**10,051**	**10,320**	**10,386**
Enteric Fermentation	5,712	5,732	5,804	5,876	6,016	6,094	6,032	5,973	5,885
Manure Management	2,613	2,708	2,801	2,990	3,283	3,447	3,567	3,861	3,990
Rice Cultivation	414	404	453	414	476	445	420	453	476
Agricultural Residue Burning	30	28	33	26	34	28	32	34	35
N_2O	**932**	**946**	**968**	**958**	**1,033**	**997**	**1,021**	**1,044**	**1,041**
Agricultural Soil Management	891	903	925	914	988	951	975	996	992
Manure Management	40	42	42	43	44	44	45	46	47
Agricultural Residue Burning	1	1	1	1	1	1	1	1	1

+ Does not exceed .5 Gg

Note: Totals may not sum due to independent rounding.

Enteric Fermentation

Methane (CH_4) is produced as part of normal digestive processes in animals. During digestion, microbes resident in an animal's digestive system ferment food consumed by the animal. This microbial fermentation process, referred to as enteric fermentation, produces methane as a by-product, which can be exhaled or eructated by the animal. The amount of methane produced and excreted by an individual animal depends primarily upon the animal's digestive system, and the amount and type of feed it consumes.

Among domestic animal types, ruminant animals (e.g., cattle, buffalo, sheep, goats, and camels) are the major emitters of anthropogenic methane because of their unique digestive system. Ruminants possess a rumen, or large "fore-stomach," in which microbial fermentation breaks down the feed they consume into soluble products that can be utilized by the animal. The microbial fermentation that occurs in the rumen enables them to digest coarse plant material that non-ruminant animals cannot. Ruminant animals, consequently, have the highest methane emissions among all animal types.

Non-ruminant domestic animals (e.g., pigs, horses, mules, rabbits, and guinea pigs) also produce anthropogenic methane emissions through enteric fermentation, although this microbial fermentation occurs in the large intestine. These non-ruminants have significantly lower methane emissions than ruminants because the capacity of the large intestine to produce methane is lower.

In addition to the type of digestive system, an animal's feed intake also affects methane excretion. In general, a higher feed intake leads to higher methane emissions. Feed intake is positively related to animal size, growth rate, and production (e.g., milk production, wool growth, pregnancy, or work). Therefore, feed intake varies among animal types as well as among different management practices for individual animal types.

Methane emissions estimates from enteric fermentation are shown in Table 5-3 and Table 5-4. Total livestock emissions in 1998 were 33.7 MMTCE (5,885 Gg). Emissions from dairy cattle remained relatively constant from 1990 to 1998 despite a steady increase in milk production. During this time, emissions per cow increased due to a rise in milk production per dairy cow (see Table 5-5); however, this trend was offset by a decline in the dairy cow population. Beef cattle emissions continued to decline, caused by the second consecutive year of declining cattle populations. Methane emissions from other animals have remained relatively constant.

Table 5-3: CH_4 Emissions from Enteric Fermentation (MMTCE)

Animal Type	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Dairy Cattle	**8.4**	**8.4**	**8.4**	**8.4**	**8.4**	**8.4**	**8.3**	**8.3**	**8.3**
Beef Cattle	**22.6**	**22.8**	**23.1**	**23.6**	**24.4**	**24.9**	**24.7**	**24.3**	**23.9**
Other	**1.6**	**1.7**	**1.7**	**1.6**	**1.7**	**1.6**	**1.6**	**1.6**	**1.6**
Sheep	0.5	0.5	0.5	0.5	0.5	0.4	0.4	0.4	0.4
Goats	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Horses	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6
Hogs	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Total	**32.7**	**32.8**	**33.2**	**33.7**	**34.5**	**34.9**	**34.5**	**34.2**	**33.7**

Table 5-4: CH_4 Emissions from Enteric Fermentation (Gg)

Animal Type	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Dairy Cattle	**1,474**	**1,465**	**1,473**	**1,468**	**1,471**	**1,473**	**1,454**	**1,453**	**1,443**
Beef Cattle	**3,951**	**3,979**	**4,039**	**4,120**	**4,256**	**4,340**	**4,305**	**4,246**	**4,165**
Other	**286**	**288**	**293**	**288**	**290**	**281**	**274**	**274**	**277**
Sheep	91	89	86	82	79	72	68	64	63
Goats	13	12	13	13	13	12	13	11	10
Horses	102	102	105	106	108	108	109	111	111
Hogs	81	85	88	87	90	88	84	88	93
Total	**5,712**	**5,732**	**5,804**	**5,876**	**6,016**	**6,094**	**6,032**	**5,973**	**5,885**

Methodology

Livestock emission estimates fall into two categories: cattle and other domesticated animals. Cattle, due to their large population, large size, and particular digestive characteristics, account for the majority of methane emissions from livestock in the United States and are handled separately. Also, cattle production systems in the United States are well characterized in comparison with other livestock management systems. Overall, emissions estimates were derived using emission factors, which were multiplied by animal population data.

While the large diversity of animal management practices cannot be precisely characterized and evaluated, significant scientific literature exists that describes the quantity of methane produced by individual ruminant animals, particularly cattle. A detailed model that incorporates this information and other analyses of feeding practices and production characteristics was used to estimate emissions from cattle populations.

To derive emission factors for the various types of cattle found in the United States, a mechanistic model of rumen digestion and animal production was applied to data on thirty-two different diets and nine different cattle types (Baldwin et al. 1987a and b).[1] The cattle types were defined to represent the different sizes, ages, feeding systems, and management systems that are typically found in the United States. Representative diets were defined for each category of animal, reflecting the feeds and forages consumed by cattle type and region. Using this model, emission factors were derived for each combination of animal type and representative diet. Based upon the level of use of each diet in the five regions, average regional emission factors for each of the nine cattle types were derived.[2] These emission factors were then multiplied by the applicable animal populations from each region.

For dairy and beef cows and replacements, emission estimates were developed using regional emission factors. Dairy cow emission factors were modified to reflect changing—primarily increasing—milk production per cow over time in each region. All other emission factors were held constant over time. Emissions from other cattle types were estimated using national average emission factors.

Emissions estimates for other animal types were based upon average emission factors representative of entire populations of each animal type. Methane emissions from these animals accounted for a minor portion of total methane emissions from livestock in the United States. Also, the variability in emission factors for each of these other animal types (e.g., variability by age, production system, and feeding practice within each animal type) is less than that for cattle.

See Annex H for more detailed information on the methodology and data used to calculate methane emissions from enteric fermentation.

Data Sources

The emission estimates for all domestic livestock were determined using a mechanistic model of rumen digestion and emission factors developed in EPA (1993). For dairy and beef cows and replacements, regional emission factors were used from EPA (1993). Emissions from other cattle types were estimated using national average emission factors from EPA (1993). Methane emissions from sheep, goats, pigs, and horses were estimated by using emission factors utilized in Crutzen et al. (1986) and annual population data from U.S. Department of Agriculture statistical reports (USDA 1994a-b, 1995a-d, 1996, 1997, 1998a-c, 1999a-i). These emission factors are representative of typical animal sizes, feed intakes, and feed characteristics in developed countries. The methodology employed in EPA (1993) is the same as those recommended in IPCC (1997). All livestock population data were taken from

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[1] The basic model of Baldwin et al. (1987a and b) was revised somewhat to allow for evaluations of a greater range of animal types and diets. See EPA (1993).

[2] Feed intake of bulls does not vary significantly by region, so only a national emission factor was derived for this cattle type.

USDA statistical reports. See the following section on manure management for a complete listing of reports cited. Table 5-5 provides a summary of cattle population and milk production data.

Table 5-5: Cow Populations (Thousands) and Milk Production (Million Kilograms)

Year	Dairy Cow Population	Beef Cow Population	Milk Production
1990	10,007	32,677	67,006
1991	9,883	32,960	66,995
1992	9,714	33,453	68,441
1993	9,679	34,132	68,328
1994	9,504	35,101	69,673
1995	9,491	35,645	70,440
1996	9,410	35,509	69,857
1997	9,309	34,629	70,802
1998	9,200	34,143	71,415

Uncertainty

The diets analyzed using the rumen digestion model include broad representations of the types of feed consumed within each region. Therefore, the full diversity of feeding strategies employed in the United States is not represented and the emission factors used may be biased. The rumen digestion model, however, has been validated by experimental data. Animal population and production statistics, particularly for beef cows and other grazing cattle, are also uncertain. Overall, the uncertainty in the emission estimate is estimated to be roughly ∀20 percent (EPA 1993).

Manure Management

The management of livestock manure can produce anthropogenic methane (CH_4) and nitrous oxide (N_2O) emissions. Methane is produced by the anaerobic decomposition of manure. Nitrous oxide is produced as part of the nitrogen cycle through the nitrification and denitrification of the organic nitrogen in livestock manure and urine.

When livestock and poultry manure is stored or treated in systems that promote anaerobic conditions (e.g., as a liquid in lagoons, ponds, tanks, or pits), the decomposition of materials in manure tends to produce methane. When manure is handled as a solid (e.g., in stacks or pits) or deposited on pastures and range lands, it tends to decompose aerobically and produce little or no methane. A number of other factors related to how the manure is handled also affect the amount of methane produced: 1) air temperature and moisture affect the amount of methane produced because they influence the growth of the bacteria responsible for methane formation; 2) methane production generally increases with rising temperature and residency time; and 3) for non-liquid based manure systems, moist conditions (which are a function of rainfall and humidity) favor methane production. Although the majority of manure is handled as a solid, producing little methane, the general trend in manure management, particularly for dairy and swine producers, is one of increasing usage of liquid systems.

The composition of the manure also affects the amount of methane produced. Manure composition varies by animal type and diet. The greater the energy content and digestibility of the feed, the greater the potential for methane emissions. For example, feedlot cattle fed a high energy grain diet generate manure with a high methane-producing capacity. Range cattle feeding on a low energy diet of forage material produce manure with roughly half the methane-producing potential of feedlot cattle manure.

The amount of N_2O produced depends on the manure and urine composition, the type of bacteria involved in the process and the amount of oxygen and liquid in the manure system. Nitrous oxide emissions result from livestock manure and urine that is managed using liquid and slurry systems, as well as manure and urine that is collected and stored as a solid. Nitrous oxide emissions from unmanaged livestock manure and urine on pastures, ranges, and paddocks, as well as from manure and urine that is spread onto fields either directly as “daily spread,” or after it is removed from manure management systems (e.g., lagoon, pit, etc.) is accounted for and discussed under Agricultural Soil Management.

Table 5-6, Table 5-7, and Table 5-8 provide estimates of methane and N_2O emissions from manure management by animal category. Estimates for methane emissions in 1998 were 22.9 MMTCE (3,990 Gg), 53 percent higher than in 1990. The majority of the increase in methane emissions was from swine and dairy cow manure and are attributed to shifts by the swine and dairy industries towards larger facilities. Larger swine and dairy farms tend to use flush or scrape liquid systems. Thus the shift towards larger facilities is translated into an increasing use of liquid systems. This shift was accounted for by incorporating weighted methane conversion factor (MCF) values calculated from the 1997 farm-size distribution reported in the *1997 Census of Agriculture* (USDA 1999m). An increase in feed consumption by dairy cows to maximize milk production is also accounted for in the estimates. A detailed description of the methodology is provided in Annex I.

Total N_2O emissions from managed manure systems in 1998 were estimated to be 4.0 MMTCE (47 Gg). The 19 percent increase in N_2O emissions from 1990 to 1998 can be partially attributed to an increase in the population of poultry and swine. The population of beef cattle in feedlots, which tend to use managed manure systems, also increased. As stated previously, N_2O emissions from unmanaged livestock manure is accounted for under Agricultural Soil Management. Methane emissions were mostly unaffected by this increase in the beef cattle population because feedlot cattle use solid storage systems, which produce little methane.

Methodology

The methodologies presented in EPA (1993) form the basis of the methane emissions estimates for each animal type. The calculation of emissions requires the following information:

- Amount of manure produced (amount per head times number of head)
- Portion of the manure that is volatile solids (by animal type)
- Methane producing potential of the volatile solids (by animal type)
- Extent to which the methane producing potential is realized for each type of manure management system (by state and manure management system)
- Portion of manure managed in each manure management system (by state and animal type)

Table 5-6: CH_4 and N_2O Emissions from Manure Management (MMTCE)

Animal Type	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
CH_4	**15.0**	**15.5**	**16.0**	**17.1**	**18.8**	**19.7**	**20.4**	**22.1**	**22.9**
Dairy Cattle	4.3	4.3	4.4	4.5	4.8	4.9	5.1	5.4	5.3
Beef Cattle	1.1	1.2	1.2	1.2	1.3	1.3	1.3	1.3	1.3
Swine	7.9	8.3	8.7	9.6	10.8	11.6	12.1	13.5	14.2
Sheep	+	+	+	+	+	+	+	+	+
Goats	+	+	+	+	+	+	+	+	+
Poultry	1.5	1.5	1.6	1.6	1.7	1.7	1.7	1.8	1.8
Horses	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
N_2O	**3.4**	**3.6**	**3.5**	**3.7**	**3.8**	**3.7**	**3.8**	**3.9**	**4.0**
Dairy Cattle	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Beef Cattle	1.4	1.6	1.5	1.5	1.6	1.5	1.5	1.5	1.6
Swine	0.1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Sheep	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Goats	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Poultry	1.6	1.7	1.7	1.8	1.8	1.9	1.9	2.0	2.0
Horses	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Total	**18.3**	**19.1**	**19.6**	**20.8**	**22.6**	**23.5**	**24.3**	**26.0**	**26.9**

+ Does not exceed 0.05 MMTCE

Note: Totals may not sum due to independent rounding.

Table 5-7: CH_4 Emissions from Manure Management (Gg)

Animal Type	1990	1991	1992	1993	1994	1995	1996	1997	1998
Dairy Cattle	747	751	762	791	843	864	896	941	933
Beef Cattle	200	205	206	212	219	221	229	229	233
Swine	1,371	1,451	1,523	1,668	1,894	2,031	2,106	2,349	2,475
Sheep	4	4	4	3	3	3	3	3	3
Goats	1	1	1	1	1	1	1	1	1
Poultry	261	268	275	284	292	297	301	308	314
Horses	29	29	30	30	31	31	31	31	31
Total	**2,613**	**2,708**	**2,801**	**2,990**	**3,283**	**3,447**	**3,567**	**3,861**	**3,990**

Note: Totals may not sum due to independent rounding.

Table 5-8: N_2O Emissions from Manure Management (Gg)

Animal Type	1990	1991	1992	1993	1994	1995	1996	1997	1998
Dairy Cattle	1.0	1.0	1.0	1.0	1.1	1.1	1.2	1.2	1.2
Beef Cattle	16.7	18.4	17.2	18.1	18.5	17.6	18.0	18.3	18.9
Swine	1.7	1.8	1.9	1.9	2.0	2.0	1.9	2.0	2.1
Sheep	0.5	0.5	0.4	0.4	0.4	0.4	0.3	0.3	0.3
Goats	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Poultry	19.1	19.8	20.4	21.0	21.7	22.3	23.0	23.5	23.9
Horses	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.8	0.8
Total	**39.8**	**42.1**	**41.7**	**43.3**	**44.4**	**44.2**	**45.3**	**46.3**	**47.3**

Note: Totals may not sum due to independent rounding.

For swine and dairy cattle —the two largest emitters of methane—estimates were developed using state-level animal population data and average weighted MCFs for each state. These weighted MCFs were determined for each farm size category based on the general relationship between farm sizes and manure system usage, where larger facilities will tend to use liquid systems. These values were further adjusted to harmonize with emissions reported in EPA (1993). For other animal types, 1990 state-level emission estimates from the detailed analysis presented in EPA (1993) were scaled by the change in the state population.

Nitrous oxide emissions were estimated by first determining manure management system usage. Manure system usage for swine and dairy cows were based on assumptions of system usage for the respective populations' farm size distribution. Total Kjeldahl nitrogen[3] production was calculated for all livestock using livestock population data and nitrogen excretion rates. Nitrous oxide emission factors specific to the type of manure management system were then applied to total nitrogen production to estimate N_2O emissions.

See Annex I for more detailed information on the methodology and data used to calculate methane emissions from manure management. The same activity data were also used to calculate N_2O emissions.

Data Sources

Annual livestock population data for all livestock types except horses were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (USDA 1994a, 1995 a-e, 1996a-b, 1997a-b, 1998a-d, 1999a-k). Horse population data were obtained from the FAOSTAT database (FAO 1999). Data on farm size distribution for dairy cows and swine were taken from the U.S. Department of Commerce (DOC 1995, 1987). Manure management system usage data for other livestock were taken from EPA (1992). Nitrogen excretion rate data were developed by the American Society of Agricultural Engineers (ASAE 1999). Nitrous oxide emission factors were

[3] Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen.

taken from IPCC/UNEP/OECD/IEA (1997). Manure management systems characterized as "Other" generally refers to deep pit and litter systems. The IPCC N_2O emission factor for "other" systems (0.005 kg N_2O/kg N excreted), was determined to be inconsistent with the characteristics of these management systems. Therefore, in its place the solid storage/drylot emission factor was used.

Uncertainty

The primary factors contributing to the uncertainty in emission estimates are a lack of information on the usage of various manure management systems in each state and the exact methane generating characteristics of each type of manure management system. Because of significant shifts in the swine and dairy sectors toward larger farms, it is believed that increasing amounts of manure are being managed in liquid manure management systems. The existing estimates reflect these shifts in the weighted MCFs based on the 1997 farm-size data. However, the assumption of a direct relationship between farm-size and liquid system usage may not apply in all cases. In addition, the methane generating characteristics of each manure management system type are based on relatively few laboratory and field measurements, and may not match the diversity of conditions under which manure is managed nationally.

The N_2O emission factors published in IPCC/UNEP/OECD/IEA (1997) were also derived using limited information. The IPCC factors are global averages; U.S.-specific emission factors may be significantly different. Manure and urine in anaerobic lagoons and liquid/slurry management systems produce methane at different rates, and would in all likelihood produce N_2O at different rates, although a single emission factor was used for both system types.

Rice Cultivation

Most of the world's rice, and all rice in the United States, is grown on flooded fields. When fields are flooded, aerobic decomposition of organic material gradually depletes the oxygen present in the soil and floodwater, causing anaerobic conditions in the soil to develop. Once the environment becomes anaerobic, methane is produced through anaerobic decomposition of soil organic matter by methanogenic bacteria. As much as 60 to 90 percent of the methane produced, however, is oxidized by aerobic methanotrophic bacteria in the soil (Holzapfel-Pschorn et al. 1985, Sass et al. 1990). Some of the methane is also leached away as dissolved methane in floodwater that percolates from the field. The remaining un-oxidized methane is transported from the submerged soil to the atmosphere primarily by diffusive transport through the rice plants. Some methane also escapes from the soil via diffusion and bubbling through floodwaters.

The water management system under which rice is grown is one of the most important factors affecting methane emissions. Upland rice fields are not flooded, and therefore are not believed to produce methane. In deepwater rice fields (i.e., fields with flooding depths greater than one meter), the lower stems and roots of the rice plants are dead so the primary methane transport pathway to the atmosphere is blocked. The quantities of methane released from deepwater fields, therefore, are believed to be significantly less than the quantities released from areas with more shallow flooding depths. Some flooded fields are drained periodically during the growing season, either intentionally or accidentally. If water is drained and soils are allowed to dry sufficiently, methane emissions decrease or stop entirely. This is due to soil aeration, which not only causes existing soil methane to oxidize but also inhibits further methane production in soils. All rice in the United States is grown under continuously flooded conditions; none is grown under deepwater conditions.

Other factors that influence methane emissions from flooded rice fields include fertilization practices (especially the use of organic fertilizers,) soil temperature, soil type, cultivar selection, and cultivation practices (e.g., tillage, and seeding and weeding practices). The factors that determine the amount of organic material that is available to decompose, i.e., organic fertilizer use, soil type, cultivar type[4], and cultivation practices, are the most important variables influencing methane emissions over an entire growing season because the total amount of methane released

[4] The roots of rice plants shed organic material. The amount of root exudates produced varies among cultivar types.

depends primarily on the amount of organic substrate available. Soil temperature is known to be an important factor regulating the activity of methanogenic bacteria, and therefore the rate of methane production. However, although temperature controls the amount of time it takes to convert a given amount of organic material to methane, that time is short relative to a growing season, so the dependence of emissions over an entire growing season on soil temperature is weak. The application of synthetic fertilizers has also been found to influence methane emissions; in particular, both nitrate and sulfate fertilizers (e.g., ammonium nitrate, and ammonium sulfate) appear to inhibit methane formation. In the United States, soil types, soil temperatures, cultivar types, and cultivation practices for rice vary from region to region, and even from farm to farm. However, most rice farmers utilize organic fertilizers in the form of rice residue from the previous crop, which is left standing, disked, or rolled into the fields. Most farmers also apply synthetic fertilizer to their fields, usually urea. Nitrate and sulfate fertilizers are not commonly used in rice cultivation in the United States. In addition, the climatic conditions of Arkansas, southwest Louisiana, Texas, and Florida allow for a second, or ratoon, rice crop. This second rice crop is produced on the stubble after the first crop has been harvested. Because the first crop's stubble is left behind in ratooned fields, the amount of organic material that is available for decomposition is considerably higher than with the first (i.e., primary) crop. Methane emissions from ratoon crops have been found to be considerably higher than those from the primary crop.

Rice cultivation is a small source of methane emissions in the United States (2 percent). Rice is cultivated in seven states: Arkansas, California, Florida, Louisiana, Mississippi, Missouri, and Texas. Estimates of total annual CH_4 emissions from rice cultivation range from 2.3 to 2.7 MMTCE (404 to 476 Gg CH_4) for the years 1990 to 1998 (Table 5-9 and Table 5-10). There was no apparent trend over the nine year period, although total emissions increased by 15 percent between 1990 and 1998 due to an increase in harvested area.

The factors that affect the rice area harvested vary from state to state.[5] In Florida, the state having the smallest harvested rice area, rice acreage is largely a function of sugarcane acreage. Sugarcane fields are flooded each year to control pests, and on this flooded land a rice crop is grown along with a ratoon crop of sugarcane (Schueneman 1997). In Missouri, rice acreage is affected by weather (e.g., rain during the planting season may prevent the planting of rice), the price differential between soybeans and rice (e.g., if soybean prices are higher, then soybeans may be planted on some of the land which would otherwise have been planted in rice), and government support programs (Stevens 1997). The price differential between soybeans and rice also affects rice acreage in Mississippi. Rice in Mississippi is usually rotated with soybeans, but if soybean prices increase relative to rice prices, then some of the acreage that would have been planted in rice, is instead planted in soybeans (Street 1997). In Texas, rice production, and thus, harvested area, are affected by both government programs and the cost of production (Klosterboer 1997). California rice area is influenced by water availability as well as government programs and commodity prices. In Louisiana, rice area is influenced by government programs, weather conditions (e.g., rainfall during the planting season), as well as the price differential between rice and corn and other crops (Saichuk 1997). Arkansas rice area has been influenced in the past by government programs. However, due to the phase-out of these programs nationally, which began in 1996, spring commodity prices have had a greater effect on the amount of land planted in rice in recent years (Mayhew 1997).

Table 5-9: CH_4 Emissions from Rice Cultivation (MMTCE)

State	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Arkansas	0.7	0.7	0.8	0.7	0.8	0.8	0.7	0.8	0.9
California	0.4	0.4	0.4	0.5	0.5	0.5	0.5	0.5	0.5
Florida	+	+	+	+	+	+	+	+	+
Louisiana	0.7	0.7	0.8	0.7	0.8	0.8	0.7	0.8	0.8
Mississippi	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Missouri	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Texas	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.3

[5] The statistic "area harvested" accounts for double cropping, i.e., if one hectare is cultivated twice in one year, then that hectare is counted as two hectares harvested.

Total	**2.4**	**2.3**	**2.6**	**2.4**	**2.7**	**2.6**	**2.4**	**2.6**	**2.7**

+ Does not exceed 0.05 MMTCE
Note: Totals may not sum due to independent rounding.

Table 5-10: CH_4 Emissions from Rice Cultivation (Gg)

State	1990	1991	1992	1993	1994	1995	1996	1997	1998
Arkansas	121.2	127	139	124	143	135	118	140	154
California	72	65	72	80	89	85	91	94	87
Florida	3	4	5	5	5	5	5	4	4
Louisiana	127	119	145	124	145	133	125	136	145
Mississippi	26	23	23	23	23	23	23	23	23
Missouri	10	12	14	12	16	14	12	15	18
Texas	55	54	55	47	55	50	47	40	44
Total	**414**	**404**	**453**	**414**	**476**	**445**	**420**	**453**	**476**

Note: Totals may not sum due to independent rounding.

Methodology

The *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) recommend applying a seasonal emission factor to the annual harvested rice area to estimate annual CH_4 emissions. This methodology assumes that a seasonal emission factor is available for all growing conditions. Because season lengths are quite variable both within and among states in the United States, and because flux measurements have not been taken under all growing conditions in the United States, an earlier IPCC methodology (IPCC/UNEP/OECD/IEA 1995) has been applied here, using season lengths that vary slightly from the recommended approach. The 1995 *IPCC Guidelines* recommend multiplying a daily average emission factor by growing season length and annual harvested area. The *IPCC Guidelines* suggest that the "growing" season be used to calculate emissions based on the assumption that emission factors are derived from measurements over the whole growing season rather than just the flooding season. Applying this assumption to the United States, however, would result in an overestimate of emissions because the emission factors developed for the United States are based on measurements over the flooding, rather than the growing, season. Therefore, the method used here is based on the number of days of flooding during the growing season and a daily average emission factor, which is multiplied by the harvested area. Agricultural extension agents in each of the seven states in the United States that produce rice were contacted to determine water management practices and flooding season lengths in each state. Although all contacts reported that rice growing areas were continuously flooded, flooding season lengths varied considerably among states; therefore, emissions were calculated separately for each state.

Emissions from ratooned and primary areas are estimated separately. Information on ratoon flooding season lengths was collected from agricultural extension agents in the states that practice ratooning, and emission factors for both the primary season and the ratoon season were derived from published results of field experiments in the United States.

Data Sources

The harvested rice areas for the primary and ratoon crops in each state are presented in Table 5-11. Data for all states except Florida for 1990 through 1995 were taken from *U.S. Department of Agriculture's National Agriculture Statistics Data—Historical Data* (USDA 1999b). The data for 1996 through 1998 were obtained from the *Crop Production 1998 Summary* (USDA 1999a). Harvested rice areas in Florida from 1990 to 1998 were obtained from Tom Schueneman (1999b, 1999c), a Florida Agricultural Extension Agent. Acreages for the ratoon crops were derived from conversations with the agricultural extension agents in each state. In Arkansas, ratooning occurred only in 1998, when the ratooned area was less than 1 percent of the primary area (Slaton 1999a). In the other three states in which ratooning is practiced (i.e., Florida, Louisiana, and Texas), the percentage of the primary area that was ratooned was constant over the entire 1990 to 1998 period. In Florida, the ratooned area was 50 percent of the primary area (Schueneman 1999a), in Louisiana it was 30 percent (Linscombe 1999a), and in Texas it was 40 percent (Klosterboer 1999a).

Information about flooding season lengths was obtained from agricultural extension agents in each state (Beck 1999, Guethle 1999, Klosterboer 1999b, Linscombe 1999b, Scardaci 1999a and 1999b, Schueneman 1999b, Slaton 1999b, Street 1999a and 1999b). These data are presented in Table 5-12.

To determine what daily methane emission factors should be used for the primary and ratoon crops, methane flux information from all the rice field measurements made in the United States was collected. Experiments in which nitrate and sulfate fertilizers, or other substances known to suppress methane formation, were applied, as well as experiments in which measurements were not made over an entire flooding season or in which floodwaters were drained mid-season, were excluded from the analysis. This left ten field experiments from California (Cicerone et al. 1992), Texas (Sass et al. 1990, 1991a, 1991b, 1992), and Louisiana (Lindau et al. 1991, Lindau and Bollich 1993, Lindau et al. 1993, Lindau et al. 1995, Lindau et al. 1998).[6] These experimental results were then sorted by season and type of fertilizer amendment (i.e., no fertilizer added, organic fertilizer added, and synthetic and organic fertilizer added). The results for the primary crop showed no consistent correlation between emission rate and type or magnitude of fertilizer application. Although individual experiments have shown a significant increase in emissions when organic fertilizers are added, when the results were combined, emissions from fields that receive organic fertilizers were not found to be, on average, higher that those from fields that receive synthetic fertilizer only. In addition, there appeared to be no correlation between fertilizer application rate and emission rate, either for synthetic or organic fertilizers. These somewhat surprising results are probably due to other variables that have not been taken into account, such as timing and mode of fertilizer application, soil type, cultivar type, and other cultivation practices. There were limited results from ratooned fields. Of those that received synthetic fertilizers, there was no consistent correlation between emission rate and amount of fertilizer applied, however, the type of synthetic fertilizer did not vary among experiments. In contrast, all the ratooned fields that received synthetic fertilizer had emission rates that were higher than the one ratoon experiment in which no synthetic fertilizer was applied. Given these results, the highest and lowest emission rates measured in primary fields that received synthetic fertilizer only—which bounded the results from fields that received both synthetic and organic fertilizers—was used as the emission factor range for the primary crop, and the lowest and highest emission rates measured in all the ratooned fields was used as the emission factor range for the ratoon crop. These ranges are 0.020 to 0.609 g/m^2-day for the primary crop, and 0.301 to 0.933 g/m^2-day for the ratoon crop.

Table 5-11: Rice Areas Harvested (Hectares)

State/Crop	1990	1991	1992	1993	1994	1995	1996	1997	1998
Arkansas									
Primary	485,633	509,915	558,478	497,774	574,666	542,291	473,493	562,525	617,159
Ratoon*	0	0	0	0	0	0	0	0	202
California	159,854	144,071	159,450	176,851	196,277	188,183	202,347	208,822	193,444
Florida									
Primary	4,978	8,580	9,308	9,308	9,713	9,713	8,903	7,689	8,094
Ratoon	2,489	4,290	4,654	4,654	4,856	4,856	4,452	3,845	4,047
Louisiana									
Primary	220,558	206,394	250,911	214,488	250,911	230,676	215,702	235,937	250,911
Ratoon	66,168	61,918	75,273	64,346	75,273	69,203	64,711	70,781	75,273
Mississippi	101,174	89,033	111,291	99,150	126,669	116,552	84,176	96,317	108,458
Missouri	32,376	37,232	45,326	37,637	50,182	45,326	38,446	47,349	57,871
Texas									
Primary	142,857	138,810	142,048	120,599	143,262	128,693	120,599	104,816	114,529
Ratoon	57,143	55,524	56,819	48,240	57,305	51,477	48,240	41,926	45,811

[6] In some of these remaining experiments, measurements from individual plots were excluded from the analysis because of the reasons just mentioned. In addition, one measurement from the ratooned fields (i.e., the flux of 2.041 g/m^2/day in Lindau and Bollich 1993) was excluded since this emission rate is unusually high compared to other flux measurements in the United States, as well as in Europe and Asia (IPCC/UNEP/OECD/IEA 1997).

Total	1,273,229	1,255,767	1,413,557	1,273,047	1,489,114	1,386,969	1,261,068	1,380,008	1,475,799

Note: Totals may not sum due to independent rounding.
* Arkansas ratooning only occurred in 1998.

Table 5-12: Rice Flooding Season Lengths (Days)

State/Crop	Low	High
Arkansas		
Primary	60	80
Ratoon	30	40
California	100	145
Florida		
Primary	90	110
Ratoon	40	60
Louisiana		
Primary	90	120
Ratoon	70	75
Mississippi	68	82
Missouri	80	100
Texas		
Primary	60	80
Ratoon	40	60

Uncertainty

The largest uncertainty in the calculation of CH_4 emissions from rice cultivation is associated with the emission factors applied. Daily average emissions, derived from field measurements in the United States, vary by more than one order of magnitude (IPCC/UNEP/OECD/IEA 1997). This variability is due to differences in cultivation practices, particularly the type, amount, and mode of fertilizer application; differences in cultivar type; and differences in soil and climatic conditions. By separating primary from ratooned areas, this Inventory has accounted for more of this variability than previous inventories. However, a range for both the primary (0.315 g/m^{2}day ±93 percent) and ratoon crop (0.617 g/m^{2}day ±51 percent) has been used in these calculations to reflect the remaining uncertainty. Based on this range, total methane emissions from rice cultivation in 1998 were estimated to have been approximately 0.43 to 5.0 MMTCE (75 to 876 Gg CH_4), or 2.7 MMTCE ±84 percent.

Another source of uncertainty is in the flooding season lengths used for each state. Flooding seasons in each state may fluctuate from year to year, and thus a range has been used to reflect this uncertainty. Even within a state, flooding seasons can vary by county and cultivar type (Linscombe 1999a).

The last source of uncertainty is in the practice of flooding outside of the normal rice season. According to the agriculture extension agents, all of the rice-growing states practice this on some part of their rice acreage, ranging from 5 to 33 percent of the rice acreage. Fields are flooded for a variety of reasons: to provide habitat for waterfowl, to provide ponds for crawfish production, and to aid in rice straw decomposition. To date, methane flux measurements have not been undertaken in these flooded areas.

As scientific understanding improves, these emission estimates will be adjusted to better reflect these variables.

Agricultural Soil Management

Nitrous oxide (N_2O) is produced naturally in soils through the microbial processes of nitrification and denitrification.[7] A number of agricultural activities add nitrogen to soils, thereby increasing the amount of nitrogen available for nitrification and denitrification, and ultimately the amount of N_2O emitted. These activities may add nitrogen to soils either directly or indirectly. Direct additions occur through various soil management practices (i.e., application of synthetic and organic fertilizers, application of sewage sludge, application of animal wastes, production of nitrogen-fixing crops, application of crop residues, and cultivation of high organic content soils, which are also called histosols), and through animal grazing (i.e., direct deposition of animal wastes on pastures, range, and paddocks by grazing animals). Indirect additions occur through two mechanisms: 1) volatilization of applied nitrogen (i.e., fertilizer, sewage sludge and animal waste) as ammonia (NH_3) and oxides of nitrogen (NO_x) and subsequent atmospheric deposition of that nitrogen in the form of ammonium (NH_4) and oxides of nitrogen (NO_x); and 2) surface runoff and leaching of applied nitrogen into aquatic systems. Figure 5-2 illustrates these sources and pathways of nitrogen additions to soils in the United States. Other agricultural soil management practices, such as irrigation, drainage, tillage practices, and fallowing of land, can affect fluxes of N_2O, as well as other greenhouse gases, to and from soils. However, because there are significant uncertainties associated with these other fluxes, they have not been estimated.

Figure 5-2: Sources of N_2O Emissions from Agricultural Soils

Estimates of annual N_2O emissions from agricultural soil management range from 75.3 to 83.9 MMTCE (891 to 992 Gg) for the years 1990 to 1998 (Table 5-13 and Table 5-14).[8] Emission levels fluctuated moderately during the 1990 to 1993 period, increased sharply in 1994, and fluctuated again through 1998. These fluctuations are largely a reflection of annual variations in synthetic nitrogen fertilizer consumption and crop production. Synthetic nitrogen fertilizer consumption, and production of corn and most beans and pulses, increased in 1994 due to the 1993 flooding of the North Central region and the intensive cultivation that followed. From 1997 to 1998, N_2O emission estimates decreased by 0.4 percent. Over the nine-year period, total emissions of N_2O increased by approximately 11 percent.

Table 5-13: N_2O Emissions from Agricultural Soil Management (MMTCE)

Activity	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Direct									
Agricultural Soils	42.7	43.3	44.7	43.0	48.3	45.3	47.1	49.3	49.2
Grazing Animals	10.3	10.3	10.6	10.7	10.9	11.1	11.0	10.7	10.5
Indirect	22.4	22.7	23.0	23.6	24.3	24.0	24.3	24.3	24.2
Total	**75.3**	**76.3**	**78.2**	**77.3**	**83.5**	**80.4**	**82.4**	**84.2**	**83.9**

Note: Totals may not sum due to independent rounding.

Table 5-14: N_2O Emissions from Agricultural Soil Management (Gg)

Activity	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Direct									
Agricultural Soils	505	512	528	509	571	536	557	583	581
Grazing Animals	121	122	125	126	129	131	130	126	124
Indirect	265	269	272	279	287	284	288	287	287

[7] Nitrification is the aerobic microbial oxidation of ammonium to nitrate, and denitrification is the anaerobic microbial reduction of nitrate to dinitrogen gas (IPCC/UNEP/OECD/IEA 1997). Nitrous oxide is a gaseous intermediate product in the reaction sequences of both processes, which leaks from microbial cells into the soil atmosphere.

[8] Note that these emission estimates include applications of N to all soils, but the phrase "Agricultural Soil Management" is kept for consistency with the reporting structure of the *Revised 1996 IPCC Guidelines.*

Total	**891**	**903**	**925**	**914**	**988**	**951**	**975**	**996**	**992**

Note: Totals may not sum due to independent rounding.

Methodology and Data Sources

This N_2O source category is divided into three components: (1) direct emissions from managed soils due to N applications and cultivation of histosols; (2) direct emissions from managed soils due to grazing animals; and (3) emissions from soils indirectly induced by applications of nitrogen. Except where specifically noted, the emission estimates for all three components follow the methodologies in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).

Direct N_2O Emissions from Agricultural Soils

Estimates of N_2O emissions from this component are based on the total amount of nitrogen that is applied to, or made available to—in the case of histosol cultivation—soils through various practices. The practices are: (1) the application of synthetic and organic fertilizers, (2) the application of sewage sludge, (3) the application of livestock and poultry waste through both daily spread and eventual application of wastes that had been managed in waste management systems (e.g., lagoons), (4) the production of nitrogen-fixing crops, (5) the application of crop residues, and (6) the cultivation of histosols.

Annual synthetic and organic fertilizer consumption data for the United States were taken from annual publications on commercial fertilizer statistics (AAPFCO 1995, 1996, 1997, 1998; TVA 1990, 1992a,b, 1994). Organic fertilizers included in these publications are manure, compost, dried blood, sewage sludge, tankage[9], and "other". The manure portion of the organic fertilizers was subtracted from the total organic fertilizer consumption data to avoid double counting[10]. Fertilizer consumption data are recorded in "fertilizer year" totals (i.e., July to June), which were converted to calendar year totals by assuming that approximately 35 percent of fertilizer usage occurred from July to December (TVA 1992b). July to December values were not available for calendar year 1998, so a "least squares line" statistical test using the past eight data points was used to arrive at an approximate total. Data on the nitrogen content of synthetic fertilizers were available in the published fertilizer reports; however, these reports did not include nitrogen content information for organic fertilizers. It was assumed that 4.1 percent of non-manure organic fertilizers on a mass basis was nitrogen (Terry 1997). Annual consumption of commercial fertilizers—synthetic and non-manure organic—in units of nitrogen are presented in Table 5-15. The total amount of nitrogen consumed from synthetic and non-manure organic fertilizers was reduced by 10 percent and 20 percent, respectively, to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997).

Data collected by the U.S. Environmental Protection Agency (EPA) were used to derive annual estimates of nitrogen additions from land application of sewage sludge. Sewage sludge is generated from the treatment of raw sewage in public or private wastewater treatment works. Based on a 1988 questionnaire returned from 600 publicly owned treatment works (POTWs), the EPA estimated that 5.4 million metric tons of dry sewage sludge were generated in the United States in that year (EPA 1993). Of this total, 36 percent was applied to land—including agricultural applications, compost manufacture, forest land application, and the reclamation of mining areas—34.0 percent was disposed in landfills, 10.3 percent was surface-disposed (in open dumps), 16.1 percent was incinerated, and 6.3 percent was dumped into the oceans (EPA 1993). In 1997, the EPA conducted a nationwide state-by-state study that estimated that approximately 7 million metric tons of dry sewage sludge were generated by 12,000 POTWs (Bastian 1999). The same study concluded that 54 percent of sewage sludge generated that year was applied to land. Sewage sludge production increased between 1988 and 1997 due to increases in the number of treatment plants and the magnitude of industrial wastewater treated, as well as changes in sewage treatment techniques. The proportion of sewage sludge applied to land increased due to the passage of legislation in 1989 that banned all ocean dumping of

[9] Tankage is dried animal residue, usually freed from fat and gelatin.

[10] The manure used in commercial fertilizer is accounted for when estimating the total amount of animal waste nitrogen applied to soils.

sewage, as well as stricter laws regulating the use of landfills for sewage disposal (Bastian 1999). To estimate sewage sludge production for the 1990 to 1998 period, the values for 1988 and 1997 were linearly interpolated. To estimate the proportion of sewage sludge that was applied to land, the values for 1988 and 1992 were linearly interpolated; the 1992 value was estimated by assuming all sewage sludge dumped in the ocean before 1992 was land applied that year (i.e., 1991 was the last year ocean dumping of sludge occurred). A second interpolation was then calculated for the period 1992 to 1997 using the 1997 value and the 1992 estimate. The rate of sewage sludge production destined for land application is currently leveling off (Bastian 1999); in the absence of more precise data for 1998, the 1997 estimate was used for 1998. Anywhere between 1 to 6 percent of dry weight sewage sludge is nitrogen, both in organic and inorganic form (National Research Council 1996); 4 percent was used as a conservative average estimate of the nitrogen content in sewage sludge. Annual land application of sewage sludge in units of nitrogen is presented in Table 5-15. As with non-manure organic fertilizer applications to managed soils, it was assumed that 20 percent of the sewage sludge nitrogen volatilizes. A portion of sewage sludge is used as commercial fertilizer; application of this nitrogen and associated N_2O emissions are accounted for under the organic fertilizer application category.

To estimate the amount of livestock and poultry waste nitrogen applied to soils, it was assumed that all of it will eventually be applied to soils with two exceptions. These exceptions are (1) the nitrogen in the poultry waste that is used as feed for ruminants (i.e., approximately 10 percent of the poultry waste), and (2) the nitrogen in the waste that is directly deposited onto fields by grazing animals.[11] Annual animal population data for all livestock types, except horses, were obtained from the USDA National Agricultural Statistics Service (USDA 1994b,c, 1995a,b, 1996a,b, 1997a,b, 1998a,b; 1999a-g,i-m). Horse population data were obtained from the FAOSTAT database (FAO 1999). Population data by animal type were multiplied by an average animal mass constant (ASAE 1999) to derive total animal mass for each animal type. Total Kjeldahl nitrogen[12] excreted per year (i.e., manure and urine) was then calculated using daily rates of nitrogen excretion per unit of animal mass (ASAE 1999) (Table 5-16). The amount of animal waste nitrogen directly deposited by grazing animals—derived using manure management system usage data and farm size (Safely et al. 1992, DOC 1995) as described in the "Direct N_2O Emissions from Grazing Animals" section—was then subtracted from the total nitrogen. Ten percent of the poultry waste nitrogen produced in managed systems and used as feed for ruminants was then subtracted. Finally, the total amount of nitrogen from livestock and poultry waste applied to soils was then reduced by 20 percent to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997).

Annual production statistics for some of the nitrogen-fixing crops (i.e., beans, pulses, and alfalfa) were taken from U.S. Department of Agriculture reports (USDA 1994a, 1997c, 1998c, 1999h). These statistics are presented in Table 5-17. Crop product values for beans and pulses were expanded to total crop dry biomass, in mass units of dry matter, by applying residue to crop ratios and dry matter fractions for residue from Strehler and Stützle (1987). Crop production for the alfalfa were converted to dry matter mass units by applying a dry matter fraction value estimated at 80 percent (Mosier 1998). To convert to units of nitrogen, it was assumed that 3 percent of the total crop dry mass for all crops was nitrogen (IPCC/UNEP/OECD/IEA 1997).

There are no published annual production statistics for non-alfalfa legumes used as forage in the United States (i.e., red clover, white clover, birdsfoot trefoil, arrowleaf clover, crimson clover, hairy vetch). Estimates of average annual crop coverage density and crop area were obtained through personal communications with agricultural extension agents or faculty at agronomy and soil science departments of universities. The estimates of dry matter crop coverage density were obtained through on-site experiment and measurement results (Smith 1999, Peterson 1999, Mosjidis 1999). Estimates of average annual crop areas at the national level are reported in Taylor and Smith (1995). Estimates of annual crop production were derived by multiplying the crop coverage densities by the crop

[11] An additional exception is the nitrogen in the waste that will runoff from waste management systems due to inadequate management. There is insufficient information with which to estimate this fraction of waste nitrogen.

[12] Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen.

areas. Total nitrogen content was estimated in the same manner as for alfalfa. Annual production estimates for non-alfalfa forage legumes are presented in Table 5-17.

To estimate the amount of nitrogen applied to soils as crop residue, it was assumed that all residues from corn, wheat, bean, and pulse production, except the fractions that are burned in the field after harvest, were either plowed under or left on the field.[13] Annual production statistics were taken from U.S. Department of Agriculture (USDA 1994a, 1997c, 1998c, 1999h). These statistics are presented in Table 5-17 and Table 5-18. Crop residue biomass, in dry matter mass units, was calculated from the production statistics by applying residue to crop mass ratios and dry matter fractions for residue from Strehler and Stützle (1987). For wheat and corn, nitrogen contents were taken from Barnard and Kristoferson (1985). For beans and pulses, it was assumed that 3 percent of the total crop residue was nitrogen (IPCC/UNEP/OECD/IEA 1997). The crops whose residues were burned in the field are corn, wheat, soybeans, and peanuts. For these crop types, the total residue nitrogen was reduced by 3 percent to subtract the fractions burned in the field (see the Agricultural Residue Burning section).

Total crop nitrogen in the residues returned to soils was then added to the unvolatilized applied nitrogen from commercial fertilizers, sewage sludge, and animal wastes, and the nitrogen fixation from bean, pulse, alfalfa and non-alfalfa forage legume cultivation. The sum was multiplied by the IPCC default emission factor (0.0125 kg N_2O-N/kg N applied) to estimate annual N_2O emissions from nitrogen applied to soils.

Statistics on the area of histosols cultivated each year were not available; however, estimates for the years 1982 and 1992 were available from *National Resources Inventory* (USDA 1994d). The area statistics for 1982 and 1992 were linearly interpolated to obtain area estimates for 1990 and 1991, and linearly extrapolated to obtain area estimates for 1993 to 1998 (Table 5-19). To estimate annual N_2O emissions from histosol cultivation, the histosol areas were multiplied by the default emission factor (8 kg N_2O-N/ha cultivated) recommended in the draft IPCC paper on "good practice" in implementing the *Revised 1996 IPCC Guidelines* (IPCC 1999a). This recommended emission factor is based on the results of recent measurements that indicate that nitrous oxide emissions from cultivated organic soils in mid-latitudes are higher than previously estimated.

Annual N_2O emissions from nitrogen applied to soils were then added to annual N_2O emissions from histosol cultivation to estimate total annual direct N_2O emissions from agricultural cropping practices (Table 5-20).

Direct N_2O Emissions from Grazing Animals

Estimates of N_2O emissions from this component were based on animal wastes that are not used as animal feed, or applied to soils, or managed in manure management systems, but instead are deposited directly on soils by animals in pastures, range, and paddocks.[14] It was assumed that all unmanaged wastes fall into this category (Safely et al. 1992), except for unmanaged dairy cow wastes. Although it is known that there is a small portion of dairy cattle that graze, there are no available statistics for this category, and therefore the simplifying assumption is made that all unmanaged dairy cow wastes fall into the daily spread category. Estimates of nitrogen excretion by the remaining animals were derived from animal population and weight statistics, information on manure management system usage in the United States, and nitrogen excretion values for each animal type.

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[13] Although residue application mode would probably affect the magnitude of emissions, a methodology for estimating N_2O emissions for these two practices separately has not been developed yet.

[14] The *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) indicate that emissions from animal wastes managed in solid storage and drylot should also be included in the emissions from soils (see footnote "c" in Table 4-22 in the Reference Manual); however, this instruction appeared to be an error (and footnote "b" should have been listed next to "Solid storage and drylot" in Table 4-22). Therefore, N_2O emissions from livestock wastes managed in solid storage and drylot are reported under Manure Management, rather than here. (See Annex H for a discussion of the activity data used to calculate emissions from the manure management source category.)

Annual animal population data for all the remaining livestock types, except horses, were obtained from the USDA National Agricultural Statistics Service (USDA 1994b,c; 1995a,b; 1996a,b; 1997a,b; 1998a,b; 1999a-g,i-m). Horse population data were obtained from the FAOSTAT database (FAO 1999). Manure management system utilization data for all livestock types except for diary cattle and swine was taken from Safely et al (1992). In the last few years, there has been a significant shift in the dairy and swine industries toward larger, consolidated facilities, which use manure management systems. Based on the assumption that larger facilities have a higher chance of using manure management systems, farm-size distribution data reported in the 1992 and 1997 Census of Agriculture (DOC 1995, USDA 1999n) were used to assess system utilization in the dairy and swine industries. Populations in the larger farm categories were assumed to utilize manure collection and storage systems; all the wastes from smaller farms were assumed to be managed as pasture, range, and paddock. As stated earlier, waste from manure collection and storage systems is covered under the manure management section. Waste from pasture, range, and paddock is considered direct depositing of waste, and is covered in this section.

For each animal type, the population of animals within pasture, range, and paddock systems was multiplied by an average animal mass constant (ASAE 1999) to derive total animal mass for each animal type. Total Kjeldahl nitrogen excreted per year was then calculated for each animal type using daily rates of nitrogen excretion per unit of animal mass (ASAE 1999). Annual nitrogen excretion was then summed over all animal types (see Table 5-21), and reduced by 20 percent to account for the portion that volatilizes to NH_3 and NO_x. The remainder was multiplied by the IPCC default emission factor (0.02 kg N_2O-N/kg N excreted) to estimate N_2O emissions (see Table 5-21).

Indirect N_2O Emissions from Nitrogen Applied to Managed Soils

This component accounts for N_2O that is emitted indirectly from nitrogen applied as commercial fertilizer, sewage sludge, and animal waste. Through volatilization, some of this nitrogen enters the atmosphere as NH_3 and NO_x, and subsequently returns to soils through atmospheric deposition, thereby enhancing N_2O production. Additional nitrogen is lost from soils through leaching and runoff, and enters groundwater and surface water systems, from which a portion is emitted as N_2O. These two indirect emission pathways are treated separately, although the activity data used are identical.

Estimates of total nitrogen applied as commercial fertilizer, sewage sludge, and animal waste were derived using the same approach as was employed to estimate the direct soil emissions. Annual application rates for synthetic and non-manure organic fertilizer nitrogen were derived from commercial fertilizer statistics as described above (AAPFCO 1995, 1996, 1997, 1998; TVA 1990, 1992a and b, 1994). Annual application rates for sewage sludge were also derived as described above. Annual total nitrogen excretion data for livestock and poultry by animal type were derived from EPA data, also as described above, using population statistics (USDA 1994b,c; 1995a,b; 1996a,b; 1997a,b; 1998a,b; 1999a-g,i-m; DOC 1987; and FAO 1999), average animal mass constants (ASAE 1999), and daily rates of nitrogen excretion per unit of animal mass (ASAE 1999). Annual nitrogen excretion was then summed over all animal types.

To estimate N_2O emissions from volatilization and subsequent atmospheric deposition, the methodology described in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) was followed, where it is assumed that 10 percent of the synthetic fertilizer nitrogen and 20 percent of animal waste (i.e., livestock and poultry) nitrogen applied as fertilizer are volatilized to NH_3 and NO_x. It was then assumed that 1 percent of the total deposited nitrogen is emitted as N_2O. The same NH_3 and NO_x volatilization and N_2O emission rates as those used for animal waste fertilizer were used for nitrogen applied to land as non-manure organic fertilizer and as sewage sludge. These emission estimates are presented in Table 5-22.

To estimate N_2O emissions from leaching and runoff, it was assumed that 30 percent of the total nitrogen applied to managed soils was lost to leaching and surface runoff, and 2.5 percent of the lost nitrogen was emitted as N_2O (IPCC/UNEP/OECD/IEA 1997). These emission estimates are also presented in Table 5-22.

Table 5-15: Commercial Fertilizer Consumption & Land Application of Sewage Sludge (Thousand Metric Tons of Nitrogen)

Fertilizer Type	1990	1991	1992	1993	1994	1995	1996	1997	1998
Synthetic	10,104	10,261	10,324	10,718	11,161	10,799	11,158	11,172	11,156

Non-Manure Organics	8	12	13	11	11	14	15	15	16
Sewage Sludge	94	103	112	120	127	135	143	151	151

Note: The sewage sludge figures do not include sewage sludge used as commercial fertilizer.

Table 5-16: Animal Excretion from Livestock and Poultry (Thousand Metric Tons of Nitrogen)

Activity	1990	1991	1992	1993	1994	1995	1996	1997	1998
Applied to Soils	3,695	3,804	3,812	3,864	3,933	3,913	3,890	3,972	3,890
Pasture, Range, & Paddock	4,830	4,850	4,972	5,021	5,132	5,221	5,170	5,029	4,923

Table 5-17: Nitrogen Fixing Crop Production (Thousand Metric Tons of Product)

Product Type	1990	1991	1992	1993	1994	1995	1996	1997	1998
Soybeans	52,416	54,065	59,612	50,885	68,444	59,174	64,780	73,176	75,028
Peanuts	1,635	2,235	1,943	1,539	1,927	1,570	1,661	1,605	1,783
Dry Edible Beans	1,469	1,532	1,026	994	1,324	1,398	1,268	1,332	1,398
Dry Edible Peas	108	169	115	149	102	209	121	264	269
Austrian Winter Peas	6	6	4	7	2	5	5	5	5
Lentils	66	104	71	91	84	97	60	108	88
Wrinkled Seed Peas	42	42	24	39	34	48	25	31	31
Alfalfa	75,671	75,585	71,795	72,851	73,787	76,671	72,137	71,887	74,398
Red Clover	62,438	62,438	62,438	62,438	62,438	62,438	62,438	62,438	62,438
White Clover	40,700	40,700	40,700	40,700	40,700	40,700	40,700	40,700	40,700
Birdsfoot Trefoil	12,375	12,375	12,375	12,375	12,375	12,375	12,375	12,375	12,375
Arrowleaf Clover	2,044	2,044	2,044	2,044	2,044	2,044	2,044	2,044	2,044
Crimson Clover	818	818	818	818	818	818	818	818	818
Hairy Vetch	500	500	500	500	500	500	500	500	500

Table 5-18: Corn and Wheat Production (Thousand Metric Tons of Product)

Product Type	1990	1991	1992	1993	1994	1995	1996	1997	1998
Corn for Grain	201,534	189,868	240,719	160,986	255,295	187,970	234,518	233,864	247,943
Wheat	74,292	53,891	67,135	65,220	63,167	59,404	61,980	67,534	69,410

Table 5-19: Histosol Area Cultivated (Thousand Hectares)

Year	Area
1990	1,013
1991	1,005
1992	998
1993	991
1994	984
1995	976
1996	969
1997	962
1998	955

Table 5-20: Direct N_2O Emissions from Agricultural Cropping Practices (MMTCE)

Activity	1990	1991	1992	1993	1994	1995	1996	1997	1998
Comm. Fertilizers & Sew. Sludge	15.2	15.5	15.6	16.2	16.9	16.3	16.9	16.9	16.9
Animal Waste	4.9	5.1	5.1	5.1	5.2	5.2	5.2	5.3	5.2
N Fixation	15.1	15.3	15.8	14.7	17.1	16.0	16.5	17.6	18.0
Crop Residue	6.4	6.3	7.1	6.0	8.0	6.8	7.5	8.4	8.1
Histosol Cultivation	1.1	1.1	1.1	1.1	1.0	1.0	1.0	1.0	1.0

Total		42.7	43.3	44.7	43.0	48.3	45.4	47.2	49.3	49.2

Note: Totals may not sum due to independent rounding.

Table 5-21: Direct N_2O Emissions from Pasture, Range, and Paddock Animals (MMTCE)

Animal Type	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Beef Cattle	9.0	9.1	9.3	9.5	9.8	10.0	10.0	9.7	9.5
Swine	0.4	0.4	0.4	0.4	0.3	0.3	0.2	0.2	0.2
Sheep	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Goats	0.1	0.1	0.1	0.1	0.1	0.1	0.1	+	+
Poultry	+	+	+	+	+	+	+	+	+
Horses	0.5	0.5	0.6	0.6	0.6	0.6	0.6	0.6	0.6
Total	**10.3**	**10.3**	**10.6**	**10.7**	**10.9**	**11.1**	**11.0**	**10.7**	**10.5**

+ Does not exceed 0.05 MMTCE

Note: Totals may not sum due to independent rounding.

Table 5-22: Indirect N_2O Emissions (MMTCE)

Activity	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Volatilization & Atm. Deposition	**3.7**	**3.7**	**3.8**	**3.8**	**3.9**	**3.9**	**4.0**	**3.9**	**3.9**
Comm. Fertilizers & Sew. Sludge	1.4	1.4	1.4	1.5	1.5	1.5	1.5	1.5	1.5
Animal Waste	2.3	2.3	2.4	2.4	2.4	2.4	2.4	2.4	2.4
Surface Run-off & Leaching	**18.7**	**19.0**	**19.2**	**19.7**	**20.4**	**20.1**	**20.4**	**20.3**	**20.3**
Comm. Fertilizer & Sew. Sludge	10.2	10.3	10.4	10.8	11.3	10.9	11.3	11.3	11.3
Animal Waste	8.6	8.7	8.8	8.9	9.1	9.2	9.1	9.0	9.0
Total	**22.4**	**22.7**	**23.0**	**23.6**	**24.3**	**24.0**	**24.3**	**24.3**	**24.2**

Note: Totals may not sum due to independent rounding.

Uncertainty

A number of conditions can affect nitrification and denitrification rates in soils. These conditions vary greatly by soil type, climate, cropping system, and soil management regime, and their combined effect on the processes leading to N_2O emissions are not fully understood. Moreover, the amount of added nitrogen from each source that is not absorbed by crops or wild vegetation, but remains in the soil and is available for production of N_2O, is uncertain. Therefore, it is not yet possible to develop statistically valid estimates of emission factors for all possible combinations of soil, climate, and management conditions. The emission factors used were midpoint estimates based on measurements described in the scientific literature, and as such, are representative of current scientific understanding. Nevertheless, estimated ranges around each midpoint estimate are wide; most are an order of magnitude or larger (IPCC/UNEP/OECD/IEA 1997; IPCC 1999a,b).

Uncertainties also exist in the activity data used to derive emission estimates. In particular, the fertilizer statistics include only those organic fertilizers that enter the commercial market, so some non-commercial fertilizer uses have not been captured. Statistics on sewage sludge applied to soils were not available on an annual basis; annual production and application estimates were based on two data points that were calculated from surveys that yielded uncertainty levels as high as 14 percent (Bastian 1999). Also, the nitrogen content of organic fertilizers varies by type, as well as within individual types; however, average values were used to estimate total organic fertilizer nitrogen consumed. Similar uncertainty levels are associated with the nitrogen content of sewage sludge. Conversion factors for the bean, pulse, alfalfa, and non-alfalfa legume production statistics were based on a limited number of studies, and may not be representative of all conditions in the United States. It was assumed that the entire crop residue for corn, wheat, beans, and pulses was returned to the soils, with the exception of the fraction burned. A portion of this residue may be disposed of through other practices, such as composting or landfilling; however, data on these practices are not available. The point estimates of yearly production yields for non-alfalfa forage legumes carry a high degree of uncertainty; many of the estimated average coverage densities and cover areas are based on a combination of on-field experimentation and expert judgment. Also, the amount of nitrogen that is added to soils from non-alfalfa forage will depend at least in part on grazing intensity, which has not been taken into account. Lastly, the livestock excretion values, while based on detailed population and weight statistics, were derived using simplifying assumptions concerning the types of management systems employed; for example, emissions due to grazing dairy cattle are probably underestimated, while emissions due to soil application of dairy cattle waste are overestimated.

Agricultural Residue Burning

Large quantities of agricultural crop residues are produced by farming activities. There are a variety of ways to dispose of these residues. For example, agricultural residues can be plowed back into the field, composted and then applied to soils, landfilled, or burned in the field. Alternatively, they can be collected and used as a fuel or sold in supplemental feed markets. Field burning of crop residues is not considered a net source of carbon dioxide (CO_2) because the carbon released to the atmosphere as CO_2 during burning is assumed to be reabsorbed during the next growing season. Crop residue burning is, however, a net source of methane (CH_4), nitrous oxide (N_2O), carbon monoxide (CO), and nitrogen oxides (NO_x), which are released during combustion.

Field burning is not a common method of agricultural residue disposal in the United States; therefore, emissions from this source are minor. The primary crop types whose residues are typically burned in the United States are wheat, rice, sugarcane, corn, barley, soybeans, and peanuts, and of these residues, less than 5 percent is burned each year, except for rice.[15] Annual emissions from this source over the period 1990 through 1998 averaged approximately 0.2 MMTCE (31 Gg) of CH_4, 0.1 MMTCE (1 Gg) of N_2O, 650 Gg of CO, and 29 Gg of NO_x (see Table 5-23 and Table 5-24).

Table 5-23: Emissions from Agricultural Residue Burning (MMTCE)

Gas/Crop Type	1990	1991	1992	1993	1994	1995	1996	1997	1998
CH_4	**0.2**	**0.2**	**0.2**	**0.1**	**0.2**	**0.2**	**0.2**	**0.2**	**0.2**
Wheat	+	+	+	+	+	+	+	+	+
Rice	+	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+	+
Corn	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Barley	+	+	+	+	+	+	+	+	+
Soybeans	+	+	+	+	0.1	+	+	0.1	0.1
Peanuts	+	+	+	+	+	+	+	+	+
N_2O	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**
Wheat	+	+	+	+	+	+	+	+	+
Rice	+	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+	+
Corn	+	+	+	+	+	+	+	+	+
Barley	+	+	+	+	+	+	+	+	+
Soybeans	0.1	0.1	0.1	+	0.1	0.1	0.1	0.1	0.1
Peanuts	+	+	+	+	+	+	+	+	+
Total	**0.3**	**0.2**	**0.3**	**0.2**	**0.3**	**0.3**	**0.3**	**0.3**	**0.3**

+ Does not exceed 0.05 MMTCE
Note: Totals may not sum due to independent rounding.

Table 5-24: Emissions from Agricultural Residue Burning (Gg)

Gas/Crop Type	1990	1991	1992	1993	1994	1995	1996	1997	1998
CH_4	**30**	**28**	**33**	**26**	**34**	**28**	**32**	**34**	**35**
Wheat	7	5	6	6	6	5	5	6	6
Rice	2	2	3	2	2	2	2	2	2

[15] The fraction of rice straw burned each year is significantly higher than that for other crops (see "Data Sources" discussion below).

Sugarcane	1	1	1	1	1	1	1	1	1
Corn	12	11	14	10	15	11	14	14	15
Barley	1	1	1	1	1	1	1	1	1
Soybeans	7	7	8	7	9	8	9	10	10
Peanuts	+	+	+	+	+	+	+	+	+
N_2O	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**
Wheat	+	+	+	+	+	+	+	+	+
Rice	+	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+	+
Corn	+	+	+	+	+	+	+	+	+
Barley	+	+	+	+	+	+	+	+	+
Soybeans	1	1	1	1	1	1	1	1	1
Peanuts	+	+	+	+	+	+	+	+	+
CO	**623**	**578**	**688**	**544**	**717**	**590**	**675**	**704**	**733**
Wheat	137	99	124	120	116	109	114	124	128
Rice	48	47	54	40	49	41	47	42	44
Sugarcane	18	20	20	20	20	20	19	21	22
Corn	254	240	304	203	322	237	296	295	313
Barley	15	16	16	14	13	13	14	13	12
Soybeans	148	153	168	144	193	167	183	207	212
Peanuts	2	3	3	2	3	2	2	2	2
NO_x	**26**	**26**	**29**	**23**	**32**	**27**	**30**	**32**	**34**
Wheat	1	1	1	1	1	1	1	1	1
Rice	1	1	2	1	1	1	1	1	1
Sugarcane	+	+	+	+	+	+	+	+	+
Corn	8	8	10	6	10	8	9	9	10
Barley	+	+	+	+	+	+	+	+	+
Soybeans	14	14	16	14	18	16	17	20	20
Peanuts	+	+	+	+	+	+	+	+	+

+ Does not exceed 0.5 Gg
Note: Totals may not sum due to independent rounding.

Methodology

The methodology for estimating greenhouse gas emissions from field burning of agricultural residues is consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). In order to estimate the amounts of carbon and nitrogen released during burning, the following equations were used:

Carbon Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned *in situ*) × (Dry Matter content of the Residue) × (Burning Efficiency) × (Carbon Content of the Residue) × (Combustion Efficiency)[16]

Nitrogen Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned *in situ*) × (Dry Matter Content of the Residue) × (Burning Efficiency) × (Nitrogen Content of the Residue) × (Combustion Efficiency)

[16] Burning Efficiency is defined as the fraction of dry biomass exposed to burning that actually burns. Combustion Efficiency is defined as the fraction of carbon in the fire that is oxidized completely to CO_2. In the methodology recommended by the IPCC, the "burning efficiency" is assumed to be contained in the "fraction of residues burned" factor. However, the number used here to estimate the "fraction of residues burned" does not account for the fraction of exposed residue that does not burn. Therefore, a "burning efficiency factor" was added to the calculations.

Emissions of CH_4 and CO were calculated by multiplying the amount of carbon released by the appropriate IPCC default emission ratio (i.e., CH_4-C/C or CO-C/C). Similarly, N_2O and NO_x emissions were calculated by multiplying the amount of nitrogen released by the appropriate IPCC default emission ratio (i.e., N_2O-N/N or NO_x-N/N).

Data Sources

The crop residues that are burned in the United States were determined from various state level greenhouse gas emission inventories (ILENR 1993, Oregon Department of Energy 1995, Wisconsin Department of Natural Resources 1993) and publications on agricultural burning in the United States (Jenkins et al. 1992, Turn et al. 1997, EPA 1992).

Crop production data were taken from the USDA's *Field Crops, Final Estimates 1987-1992, 1992-1997* (USDA 1994, 1998) and *Crop Production 1998 Summary* (USDA 1999), except data on the production of rice in Florida, which USDA does not estimate. To estimate Florida rice production, an average 1998 value for ice productivity (i.e., metric tons rice/acre) was obtained from Sem-Chi Rice, which produces the majority of rice in Florida (Smith 1999), and multiplied by total Florida rice acreage each year (Schueneman 1999c). The production data for the crop types whose residues are burned are presented in Table 5-25.

The percentage of crop residue burned was assumed to be 3 percent for all crops in all years, except rice, based on state inventory data (ILENR 1993, Oregon Department of Energy 1995, Noller 1996, Wisconsin Department of Natural Resources 1993, and Cibrowski 1996). Estimates of the percentage of rice acreage on which residue burning took place were obtained on a state-by-state basis from agricultural extension agents in each of the seven rice-producing states (Guethle 1999, Fife 1999, Klosterboer 1999a and 1999b, Slaton 1999a and 1999b, Linscombe 1999a and 1999b, Schueneman 1999a and 1999b, Street 1999a and 1999b) (see Table 5-26 and Table 5-27). The estimates provided for each state remained the same from year to year for all states, with the exception of California. For California, it was assumed that the annual percents of rice acreage burned in Sacramento Valley are representative of burning in the entire state, because the Valley accounts for over 95 percent of the rice acreage in California (Fife 1999). The annual percents of rice acreage burned in Sacramento Valley were obtained from Fife (1999). These values declined over the 1990-1998 period because of a legislated reduction in agricultural burning (see Table 5-27). Because the percentage of rice acreage burned varied from state to state, and from year to year within California, a weighted average national "percent burned" factor was derived for rice for each year (Table 5-27). The weighting was based on rice area in each state.

Residue/crop product mass ratios, residue dry matter contents, residue carbon contents, and residue nitrogen contents for all crops except sugarcane, peanuts, and soybeans were taken from Strehler and Stützle (1987). These data for sugarcane were taken from University of California (1977) and Turn et al. (1997). Residue/crop product mass ratios and residue dry matter contents for peanuts and soybeans were taken from Strehler and Stützle (1987); residue carbon contents for these crops were set at 0.45 and residue nitrogen contents were taken from Barnard and Kristoferson (1985). The value for peanuts was set equal to the soybean value. These assumptions are listed in

Table 5-28. The burning efficiency was assumed to be 93 percent, and the combustion efficiency was assumed to be 88 percent for all crop types (EPA 1994). Emission ratios for all gases (see Table 5-29) were taken from the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).

Table 5-25: Agricultural Crop Production (Thousand Metric Tons of Product)

Crop	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**
Wheat	74,292	53,891	67,135	65,220	63,167	59,404	61,980	67,534	69,410
Rice	7,105	7,271	8,196	7,127	9,019	7,935	7,828	8,339	8,570
Sugarcane	25,525	27,444	27,545	28,188	28,057	27,922	26,729	28,766	30,588
Corn[a]	201,534	189,868	240,719	160,986	255,295	187,970	234,518	233,864	247,943
Barley	9,192	10,110	9,908	8,666	8,162	7,824	8,544	7,835	7,674
Soybeans	52,416	54,065	59,612	50,885	68,444	59,174	64,780	73,176	75,028
Peanuts	1,635	2,235	1,943	1,539	1,927	1,570	1,661	1,605	1,783
Total	**371,698**	**344,883**	**415,058**	**322,612**	**434,069**	**351,799**	**406,041**	**421,120**	**440,995**

[a]Corn for grain (i.e., excludes corn for silage).

Table 5-26: Percentage of Rice Area Burned By State

State	Percent Burned
Arkansas	10
California	variable[a]
Florida[b]	0
Louisiana	6
Mississippi	10
Missouri	3.5
Texas	2

[a]Values provided in Table 5-27.
[b]Burning of crop residues is illegal in Florida.

Table 5-27: Percentage of Rice Area Burned

Year	California	U.S. (weighted average)
1990	43	12
1991	43	12
1992	43	12
1993	26	10
1994	24	10
1995	20	9
1996	27	11
1997	16	9
1998	19	9

Table 5-28: Key Assumptions for Estimating Emissions from Agricultural Residue Burning[a]

Crop	Residue/ Crop Ratio	Fraction of Residue Burned	Dry Matter Fraction	Carbon Fraction	Nitrogen Fraction
Wheat	1.3	0.03	0.85	0.4853	0.0028
Rice	1.4	variable[b]	0.85	0.4144	0.0067
Sugarcane	0.8	0.03	0.62	0.4235	0.0040
Corn	1.0	0.03	0.78	0.4709	0.0081
Barley	1.2	0.03	0.85	0.4567	0.0043
Soybeans	2.1	0.03	0.87	0.4500	0.0230
Peanuts	1.0	0.03	0.90	0.4500	0.0230

[a] The burning efficiency and combustion efficiency for all crops were assumed to be 0.93 and 0.88, respectively.
[b] See Table 5-27.

Table 5-29: Greenhouse Gas Emission Ratios

Gas	Emission Ratio
CH_4[a]	0.005
CO[a]	0.060
N_2O[b]	0.007
NO_x[b]	0.121

[a] Mass of carbon compound released (units of C) relative to mass of total carbon released from burning (units of C)
[b] Mass of nitrogen compound released (units of N) relative to mass of total nitrogen released from burning (units of N)

Uncertainty

The largest source of uncertainty in the calculation of non-CO_2 emissions from field burning of agricultural residues is in the estimates of the fraction of residue of each crop type burned each year. Data on the fraction burned, as well as the gross amount of residue burned each year, are not collected at either the national or state level. In addition, burning practices are highly variable among crops, as well as among states. The fractions of residue burned used in these calculations were based upon information collected by state agencies and in published literature. It is likely that these emission estimates will continue to change as more information becomes available in the future.

Other sources of uncertainty include the residue/crop product mass ratios, residue dry matter contents, burning and combustion efficiencies, and emission ratios. A residue/crop product ratio for a specific crop can vary among cultivars, and for all crops except sugarcane, generic residue/crop product ratios, rather than ratios specific to the United States, have been used. Residue dry matter contents, burning and combustion efficiencies, and emission ratios, all can vary due to weather and other combustion conditions, such as fuel geometry. Values for these variables were taken from literature on agricultural biomass burning.

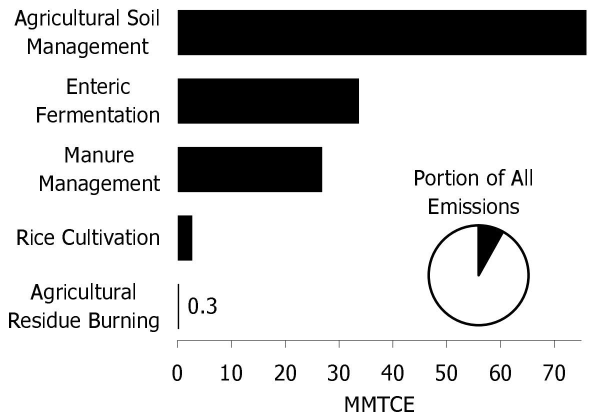

Figure 5-1: 1998 Agriculture Chapter GHG Sources

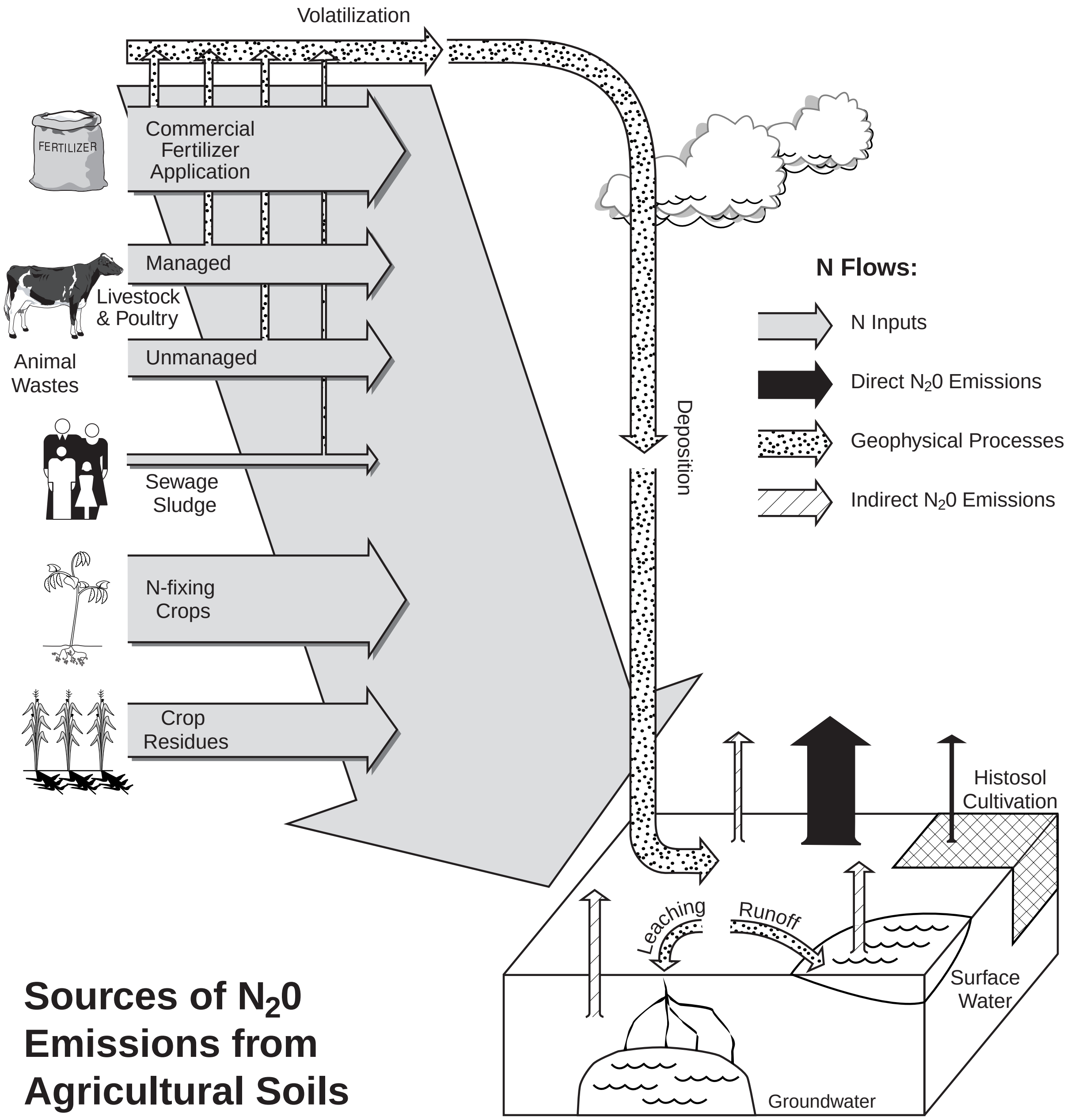

Fig. 5.2 This graphic illustrates the sources and pathways of nitrogen that result in direct and indirect N_2O emissions from agricultural soils in the United States. Nitrogen flows through the system are represented by arrows, with the width of each arrow proportional to the magnitude of the nitrogen flow; however, nitrogen inputs are not strictly proportional in magnitude, relative to N_2O emissions, as represented here. Sources of nitrogen applied to, or deposited on, soils are represented on the left-hand side of the graphic. Note that volatilization occurs during or after the application of N to soils. Geophysical processes such as volatilization, deposition, leaching, and runoff, actively circulate N in various chemical forms through the atmosphere, soils, and aquatic systems. On the lower right-hand side is a cut-away view of a representative section of a managed soil; histosol cultivation is represented here.